AQUA-3: Forestry Impacts on Water Quality

Stephanie Fulton and Ben West

Region 4, US Environmental Protection Agency

How have forest management activities and other forest uses influenced water quality, aquatic habitat, and designated uses in forested watersheds?

1 Key Findings

- In the absence of controlling measures such as Best Management Practices (BMPs), silvicultural operations have the potential to significantly impact general water quality by generating nonpoint source pollution.

- From 1988 to 1998, an annual average of approximately 3,600 miles of rivers and streams were considered potentially impaired by pollution from silvicultural activities throughout the South.

- When compared with other land uses in the South, silvicultural activities are consistently found to be minor nonpoint sources of water-quality impacts (see Chapter AQUA-1). Silviculture was one of the lowest "leading sources" of pollution or impairment for rivers and streams between 1988 and 1998 as reported by Southern States.

- BMPs, when appropriately implemented and maintained, are very effective in controlling nonpoint sources of pollution. They are particularly important in areas with steep topography.

- Most impacts are short-term (first several years after harvest), decreasing over time as vegetation regrows.

- The major potential nonpoint-source impact resulting from silvicultural activities is sediment from roads and skid trails. Other minor nonpoint- source impacts on water quality include short-term increased peak flows during storms, short-term increased base flows, short-term increased nutrient concentrations (primarily nitrogen and phosphorous), short-term increases in herbicides, fertilizers and derivative products, and thermal pollution (increased stream temperature).

- There is very little information available on the cumulative effects of past and ongoing timber harvesting on overall watershed health.

2 Introduction

The quality of water draining forested watersheds is typically the highest in the country (Brown and Binkley 1993, Clark and others 2000). For this reason, the effects of forestry activities on water quality have been widely studied (Brown and Binkley 1994, National Council for And Stream Improvement 1994, National Council for And Stream Improvement 1999, Riekerk and others 1989, Stickney and others 1994, Swank and others 1989). It has been found that pollution impacts on water quality from forestry activities are generally local in nature, short-lived, less frequent, and are less extensive in nature than activities related to either agricultural or urban activities (Bethea 1985, Dissmeyer 2000). For a complete discussion on various types and sources of pollution and the relative impacts of silvicultural versus other land-use activities on water quality in the South see Chapter AQUA-1. Chapter SOCIO-3 describes the many laws and regulations governing silvicultural nonpoint-source impacts on water quality.

Without adequate controls, however, forestry operations do have the potential to significantly affect high-quality water sources and critical fisheries habitat. Silvicultural operations that can cause nonpoint-source pollution include road and skid trail construction, tree cutting and removal, site preparation and stand regeneration treatments, herbicide application, fertilizer application, and prescribed burning. The major types of potential pollutants produced by these sources include sediment, logging equipment fluids, nutrients from harvested areas and applied fertilizers, forestry pesticides, and increased water temperature or thermal pollution.

This Chapter describes how forest management activities and pollutants influence water quality. Prior to the enactment of the Clean Water Act (CWA) in 1972, research on forest water quality examined the impacts of forestry activities characterized by the absence of controls over how and where trees were cut or how they were removed. Since that time, however, water-quality research has begun to focus on the effectiveness of best management practices (BMPs) for maintaining water quality while harvesting trees. In response to the CWA, there is a growing body of research on the effectiveness of BMPs in protecting water quality. Chapter AQUA-4 specifically describes the range of appropriate silvicultural BMPs and addresses the effectiveness of BMPs in protecting water quality in the south.

While there is a considerable amount of overlap between Chapters TIMBR-3, AQUA-1, AQUA-2, AQUA-4, and AQUA-5, this Chapter focuses specifically on the impacts of silvicultural activities on water. From public meetings and written comments obtained when the Assessment was being planned, a list was compiled of major points to address in this Chapter. These included:

- Evaluate how these activities have, and can influence hydrologic response.

- Include a consideration of all relevant water quality parameters: biological, chemical and physical.

- Examine effects of pesticides, sediment, and fertilizer.

- Examine the influence of these activities on municipal water supplies.

- Discuss how impacts may differ depending on the size and intensity of harvest and other treatments.

- Identify any differences in water quality impacts of hardwood versus pine management and plantations versus natural stands.

Each of these items is discussed in the Results section of this Chapter, with the exception of the influence of forestry activities on municipal water supplies and designated uses (for a definition of designated uses see Chapter AQUA-1). Specific information on these topics was not identified during research conducted for this Chapter. However, the impacts of individual water quality pollutants, including sediment, nutrients, pesticides/herbicides, resulting from forestry activities on designated uses, such as drinking water supply, primary contact recreation, or wildlife habitat, are generally discussed in sections related to individual pollutants.

3 Methods and Data Sources

Existing literature, which is extensive, was reviewed to describe impacts from silvicultural activities on water quality. Given the magnitude of the study area and the generally localized nature of water-quality impacts from silviculture, the primary objective for this Chapter was to compile an extensive, current summary of literature on the subject. No original research was conducted.

Primary data sources include Federal agency reports, academic and professional journals, and workshop proceedings. An attempt was made to: (1) identify the most recent literature on the subject matter, and (2) identify appropriate references and studies that have been completed across the entire 13-State study area.

4 Results

Brown and Binkley (1994) compiled an extensive review of land management impacts on water quality in North America. They concluded that there is the potential for forestry operations to adversely affect water quality if BMPs are poorly implemented. Without adequate controls, forestry operations may degrade several water-quality characteristics in waterbodies receiving drainage from forests (Mostaghimi and others 1999). Sediment concentrations can increase due to accelerated erosion; water temperatures can increase due to removal of overstory riparian shade; slash and other organic debris can accumulate in waterbodies, depleting dissolved oxygen; and organic and inorganic chemical concentrations can increase due to harvesting and fertilizer and pesticide applications (Brown 1985). These potential increases in contaminants are usually proportional to the severity of site disturbance (Riekerk 1985, Riekerk and others 1989). Impacts of silvicultural nonpoint-source pollution depend on site characteristics, climatic conditions, and the forest practices employed.

The U.S. Environmental Protection Agency (EPA) publishes a biennial national assessment of water quality, summarizing State reports that are based on monitoring, surveys of scientists, water-quality modeling, and citizen input. EPA National Water Quality Inventory Reports from 1988 to 1998 reported an annual average of approximately 3,600 miles of rivers and streams that were considered potentially impaired by nonpoint-source pollution from silviculture activities throughout the South (Table 1). An impaired water is defined as any waterbody that is classified as partially supporting, or not supporting, its designated use(s) (see Chapter AQUA-1). From 1988 to 1998, Mississippi reported the greatest average number of river and stream miles per year (1,216 miles) that were considered impaired by forestry activities, followed by Louisiana (984 miles) and Florida (563 miles). Texas did not report any river and stream miles as being impaired by forestry activities during this time frame. Georgia reported an average of one river and stream mile per year as being impaired by silvicultural activities.

The information displayed in Table 1 represents an aggregation of current, localized water-quality problems that have been partially or wholly attributed to silvicultural activities and reported by individual States. Given the magnitude of the study area, it was not possible to identify and summarize the extent of these localized problems for this report.

Table 1 highlights the extreme variability in river and stream miles impaired by silvicultural activities as identified by States. Because of this variability, the National Association of State Foresters (NASF) and the Society of American Foresters (SAF) conducted a thorough review of waterbodies listed as impaired by silvicultural operations (Society of American Foresters 2000). In their review, they concluded that EPA and the States overestimated the amount of waters affected by silviculture. The study cited two major problems with the listing process: (1) inconsistent data reporting and (2) insufficient water-quality data. There is a great deal of interstate variability in how State reports are compiled. For example, some States may simply identify silviculture as a general source of nonpoint-source pollution; other States may distinguish between different silvicultural operations such as road building, site preparation, herbicide application, etc (Society of American Foresters 2000). In addition, some listings are a result of deforestation rather than silviculture. An instance is cited in Louisiana where the actual cause of impairment was deforestation for residential development rather than forestry operations.

Clearly there is uncertainty regarding the accuracy of State listings of impairment due to silviculture. Despite these limitations, this information represents the most comprehensive set of current water-quality data available for the South. These reports were used in this Chapter to identify general trends over time at the regional and State level. A more thorough discussion of the EPA National Water Quality Inventory Reports and the relative importance of silviculture as a source of water-quality problems is included in Chapter AQUA-1.

The major impacts of silvicultural activities on water quality described here are: (1) changes in hydrological responses of watersheds, (2) increases in sedimentation, (3) increases in temperature, (4) reductions in dissolved oxygen content, (5) increases in nutrient content of streams, (6) effects on aquatic habitat and biota, and (7) effects on forested wetlands. A final section addresses the water-quality effects associated with silvicultural management intensity and specific site-preparation techniques, such as fertilizer or herbicide application and prescribed burning.

4.1 Hydrologic Response

Seven processes are at work in the terrestrial portion of the hydrologic cycle:condensation, precipitation, interception, infiltration, surface runoff, subsurface flow, and evapotranspiration. These occur simultaneously and, except for precipitation, continuously. Precipitation begins after water vapor becomes too heavy to remain in atmospheric air currents. During rainfall, some precipitation is caught on vegetative surfaces, and the water may evaporate before reaching the ground surface. This is moisture called interception. A portion of the precipitation that reaches the Earth's surface seeps into the ground through the process called infiltration. The amount of water that infiltrates the soil varies with rainfall intensity, the degree of land slope, the amount and type of vegetation, the soil and rock type, and whether the soil is already saturated with water. The more openings in the surface (cracks, pores, joints), the more infiltration occurs. Precipitation that reaches the surface of the Earth but does not infiltrate into the soil is called surface runoff. When there is a lot of precipitation, soil may become saturated with water, and additional rainfall can no longer enter it. Surface runoff will quickly drain into creeks, streams, and rivers, adding a large amount of water to their flow. Along the way, some water evaporates, percolates into the ground, or is used for agricultural, residential, or industrial purposes. The infiltrated water either moves by subsurface pathways to the stream system, or it is taken up by plants through their roots and transpired. Evapotranspiration is water evaporating from the ground and transpiring from plants, or the total water vapor added to the atmosphere.

Streamflow is water moving through a stream channel and is comprised of both baseflow and stormflow. Between storm events, streamflow is dominated by baseflow resulting from soil moisture and groundwater discharge to the channel (Hewlett 1961, Hewlett and Hibbert 1966). During and shortly after a storm, streamflow rises and then falls back toward baseflow conditions. Such pulses of water during storms are called stormflow. Changes in flows attributed to forestry activities (especially timber removal) are generally measured as an average change in inches of surface runoff, and then reported relative to a control watershed. Peakflow is the maximum flow rate that occurs in a specified period of time, usually across a year or during a storm. A perennial stream is one which flows throughout the year. Intermittent and ephemeral streams flow seasonally and during storms, respectively.

Silvicultural activities can impact the hydrologic cycle by affecting soil compaction, amount of vegetative soil cover, evapotranspiration, infiltration into soil, interception loss, soil moisture, and snow melt/accumulation (Reid 1993). Timber removal can drastically change interception amounts for several years after harvest and temporarily alter the water balance of a watershed by reducing total evapotranspiration. In general, reduced evapotranspiration rates result in higher soil moisture, groundwater recharge, and streamflow (Ursic and Douglas 1979). Reduced evapotranspiration rates can also cause increased stormflows because soils are wetter at the start of each rainfall event. Increases in surface runoff can be attributed to many different factors, including amount of precipitation, antecedent climatic conditions such as drought, hurricanes, percent of timber removed, soil compaction, infiltration, and soil moisture.

Hydrologic changes after a timber harvest usually include increases in total water yield (baseflow plus stormflow), increases in total streamflow, and higher water tables (Ursic and Douglas 1979, Likens and others 1970, Douglas and Helvey 1971, Lynch and Corbett 1990, Riekerk 1985, Mostaghimi and others 1999), and increases in total amount and timing of storm runoff and peakflow rates (Swank and others 1988, Beasley and Granillo 1988, Blackburn and others 1986, Ursic 1991, Van Lear and others 1985, Mostaghimi and others 1999).

Forestry activities can also impact hydrologic regimes by altering the land’s topography. For example, tractor skid trails can channel and concentrate erosive flows. Shallow subsurface flows can also be influenced by the use of mechanical equipment for site preparation and planting activities (Scoles and others 1996, Mostaghimi and others 1999). Minor drainage interruptions can occur when skid trails or road construction redirect flows from one drainage to another (Reid 1993).

4.1.1 Baseflow and Stormflow

Numerous studies have demonstrated increased water yields in the form of both increased baseflow and stormflow in response to timber cutting. Increased baseflows and stormflows can increase channel scouring, erosion, and downstream deposition of eroded materials. Streamflow increases are approximately proportional to the percentage of trees removed (Patric 1978). Maximum increases in water yield result from clearcutting and extensive site preparation, which completely remove vegetation. Flows return to normal levels within several years as vegetation regrows (Hibbert 1966, Swank and others 1988, Swift and Swank 1981, Bosch and Hewlett 1982, Scoles and others 1996).

Rice and Wallis (1962) found that streamflow increased 2.08 inches relative to an undisturbed control watershed after the harvest of 2.8 million board feet and the construction of approximately 3 miles of new logging roads in a 4-square-mile watershed in the Sierra Nevada mountains in California. In a worldwide survey of the literature on timber harvesting and water yield, Bosch and Hewlett (1982) found that cutting 10 percent of the pine forest on a watershed increased annual stormflow by approximately 1.6 inches in the first year after harvest. Harvesting 100 percent of the watershed increased flows between 7 and 20 inches during the first year after harvest.

Scoles and others (1996) reported that annual stormflows increased an average of 4 inches off both clearcut and selectively cut watersheds in the year after harvest in Arkansas compared to an uncut watershed. The increase was not statistically significant, however, due to the variability in stormflows between watersheds. In a different large-scale watershed study in Arkansas, Scoles and others (1996) found that average annual streamflow (corrected for rainfall) increased by 20 percent (3.9 inches) after 20 percent of the watershed was converted to pine plantation less than 10 years old, accompanied by a rapid expansion of the road network. Most of the total increase was seen during the dormant season (October-February). This increase in streamflow after conversion of hardwood forest to planted pine contrasts with the more usual result of decreased flows following conversion to pine (Swank and Miner 1968, Swank and Douglas 1974) (see Section 4.9 Hardwood Conversion to Planted Pine). This contrast may be due to the fact that the plantations described by Scoles and others (1996) were generally less than 10 years old and not transpiring at their maximum possible rate.

Lebo and Herrman (1998) examined outflow characteristics from 1986 to 1994 in a low-level pocosin site with artificial drainage in an 1,161-acre watershed on the Coastal Plain of North Carolina. They evaluated effects of semiannual road maintenance, timber harvest, site preparations, and replanting on water quality. Approximately 60 percent of the site was harvested during the study period. BMPs for the State of North Carolina were implemented where applicable. Although comparison of harvest and nonharvest years was complicated by variations in annual rainfall, the authors found that a 47 percent increase in outflow (4.33 to 6.40 inches) was associated with the harvesting of trees. The effects persisted for a year after the sites were prepared for planting.

There have been several exceptions reported in the literature where average annual stormflow on clearcut sites actually decreased following intensive site preparation compared to a control site (Mostaghimi and others 1999, Scoles and others 1996). Scoles and others (1996) reported decreased average annual stormflow in the first year after clearcutting and intensive site preparation on watersheds in Oklahoma. Those authors hypothesized that the unexpected decreases may have been due to "subsoiling," a site-preparation method similar to deep plowing that creates soil furrows and often destroys soil texture, sealing large macropores created by old root channels, animal burrows, or soil cracks. Sealing of these macropores allows for collection of rainwater in the soil furrows with less draining of stormwater to ephemeral stream channels.

Similarly, Mostaghimi and others (1999) found that storm-runoff volumes were reduced after clearcutting and site preparation on sites in the Virginia Coastal Plain both with and without BMPs. The authors attributed the reduction of flows to the disruption of subsurface flow pathways from soil compaction and site-preparation activities similar to subsoiling.

Scoles and others (1996) found that the increases in stormflow off clearcut and selectively cut watersheds were greater than those off control watersheds during low-flow periods, primarily the growing season and fall. The increase in stormflow from harvested watersheds during the growing season is particularly evident because of the lack of water uptake from vegetation. The lack of vegetation often leads to soil saturation and, subsequently, greater volumes of water entering the stream system (Scoles and others 1996).

4.1.2 Peakflows

Research has generally concluded that forest harvesting has little influence on the size of a major peakflow (Hewlett and Helvey 1970). Scoles and others (1996) found that while peakflows increased with harvest intensity in several small watersheds in the Ouachita mountains in Arkansas, the differences were not statistically significant. After large storms, peakflows did not differ much between an undisturbed watershed and a harvested area. They speculated that when there is significant rainfall and the soil is saturated with moisture, presence of vegetation in the watershed has less of an effect on mitigation of peakflows (Scoles and others 1996). However, soil and geologic features can produce wide variations in peakflows.

4.2 Sedimentation

Many studies have shown that the most important water-quality problem associated with forestry activities is sedimentation. Harvest and site preparation techniques that expose bare soil to the erosional influence of raindrops have the greatest potential to impact water quality. Areas where soil has been disturbed are subject to erosion, resulting in the downslope movement of sediment after it rains. The movement of sediment downhill is related to the steepness of the slope and soil erodibility (National Council for Air and Stream Improvement 1994). Soil erodibility greatly influences the magnitude of soil erosion and transport. Factors that affect soil erodibility include soil texture, percent organic matter, presence of a litter layer, infiltration rate, and bulk density. Sources of sediment include roads and ditches (particularly at stream crossings), bare soil on steep slopes, cutbanks, slope failures and debris flows, and streambank erosion and channel scour. For a more complete discussion of the factors influencing soil erosion and sedimentation see Chapter AQUA-4.

Fine sediments can impair habitat primarily by: (1) reducing the permeability of streambed gravels, which reduces water and gas exchange; (2) burying gravels, which inhibits or prevents the movement of organisms and materials between the stream channel and the river-influenced groundwater zone; (3) filling pools (National Council for Air and Stream Improvement 1994); and (4) covering salmonid nests, which prevents emergence and survival of fish fry (Waters 1995). Most timber harvest impacts are related to the access and movement of vehicles and machinery, and the skidding and loading of trees or logs. Simply felling trees does not accelerate erosion much above geologic rates (Patric 1978). It does not compact the soil and initially it actually adds to the litter layer. Revegetation is so quick in eastern hardwood forests that vegetation covers the soil within 2 to 3 years.

Harvesting activities that have the greatest erosion potential include the construction and use of haul roads, skid trails, and landings for access to and movement of logs, particularly in areas with steep slopes (Brown and Binkley 1994, National Council for Air Stream Improvement 1994, Patric 1978). Site preparation with large tractors that shear, disk, drum-chop, or root-rake a site usually result in considerable soil disturbance and compaction. Extensive vehicle movement removes vegetation and litter cover, which exposes and disturbs bare mineral soil. Harvesting and site preparation activities can also create furrows and depressions that can capture and hold eroded soil. Decreased infiltration and percolation of precipitation may result in increased stormflows and runoff with high erosive forces. The retention of logging slash protects bare soil by intercepting rainfall, minimizing soil detachment.

Sedimentation impacts from forestry operations are generally short-lived. Major impacts occur during and for several years after road construction activities -- until road surfaces and cut and fill slopes stabilize. In examining the effects of logging on streamflow and sedimentation in a California watershed in the Sierra Nevada, Rice and Wallis (1962) reported that suspended sediment increased 8-fold (from 0.25 tons/acre to 4.12 tons/acre) in the first year after logging and dropped to 0.47 tons/acre, or twice its normal rate, by the second year.

Forestry professionals now commonly recognize that roads and skid trails are the major sources of sediment from forestry-related activities (Brown and Binkley 1994, Patric 1976, Swift 1988, Swift 1984a, 1984b, USDA Forest Service 1984, Yoho 1980,). Scoles and others (1996) report that up to 90 percent of stream sediment following timber harvesting is road-related. Skidding logs across the forest floor exposes and compacts mineral soil, increasing chances of overland flow. Without overland flow there is no mechanism to carry detached soil particles to stream channels. Skidding many logs along the same track creates furrows that tend to channel and increase the erosive force of overland flows.

The Coweeta Hydrologic Research Laboratory began a series of watershed treatments in the 1940's to demonstrate the effects of timber harvesting on soil loss and water quality. These early studies emphasized the importance of roads and skid trails as sources of sediment to surface waters. Lieberman and Hoover (1948) reported that average stream turbidities during this Coweeta logging demonstration without best management practices were 96 parts per million (ppm), with a maximum turbidity level of 5,700 ppm during a storm in 1947. Typical logging practices in this era included steep access roads and skid trails constructed parallel and adjacent to streams. No controls were used to protect water quality. A control watershed exhibited average turbidities of 4.3 ppm with a maximum turbidity of 80 ppm. A second demonstration with extensive BMPs showed, by contrast, the value of erosion control practices (Dils 1957, Swift 1988).

Beasley and others (1984) related sediment loss associated with forest roads to the average slope gradient of road segments. The greater the average slope gradient, the greater the soil loss, ranging from a total of 6.8 tons/acre lost when the slope gradient was 1 percent, to 19.4 tons/acre at 4 percent, to 32.3 tons/acre at 6 percent, to 33.7 tons/acre at 7 percent. In addition, soil loss from roadbeds occurs primarily during the short period immediately after construction, but before the roadbed is completed and grass seed has become well established. In studies testing road design guidelines, one study found that three-quarters of the eroded soil was washed into the stream immediately below a road crossing during the first 2 months of the study; another 15 percent was measured a year later during the 3-month period when the road was being used for hauling logs (Swift 1988).

Road crossings over defined channels are the most critical points on a road system because fills are larger, the road drains directly into the stream system, and opportunities for mitigating practices are limited. Roadside ditches can also be a particularly large and direct source of sediment into streams and rivers (Reid and Dunne 1984, Sullivan and Duncan 1981). Spacing between drainage structures should decrease as slope increases to reduce the erosive power of ditch water (Scoles and others 1996, Swift and Burns 1999).

Careful location and layout of roads and logging operations can greatly affect the magnitude of sediment. Limiting equipment operation and construction of roads, skid trails, and landings also reduces the amount of sediment entering streams (Rice and Wallis 1962, Stringer and Thompson 2000). In an overview of road construction studies conducted at Coweeta, Swift (1988) describes the various components of road construction activities and compares their impacts. These studies developed improved road building techniques and other logging practices and demonstrate that logging roads could be built in the Appalachian Mountains without reducing water quality. In fact, current BMP guidelines for forest access roads are, "almost without exception," based on Coweeta experience (Swift 1988). Soil loss can be reduced by up to 50 percent through proper planning and use of BMPs (Scoles and others 1996, Yoho 1980). A more thorough discussion of sediment impacts from roads and applicable BMPs is included in Chapter AQUA-4.

4.3 Temperature

Many factors affect stream temperature, including incoming solar radiation, evaporation rates, topography, height and density of vegetation, amount of streamflow, depth and direction of flow, and temperature of water entering streams from subsurface flow (National Council for Air and

Stream Improvement 1994, Scoles and others 1996). Forest practices may impact stream temperatures through: (1) the removal of streamside forest canopy, (2) alteration of the size and shape of stream channels, and (3)change in the volume of low flows (National Council For Air And Stream Improvement 1994). Increased temperatures in streams and waterbodies can result from vegetation removal in the riparian zone. Aquatic organisms have adapted to seasonal variations in temperature, but temperature increases due to vegetation removal can be dramatic in small streams, adversely affecting aquatic species and habitat (Brown 1972, Curtis and others 1990, Megahan 1980). Increased water temperatures can: (1) reduce the amount of dissolved oxygen that a stream or waterbody can absorb, (2) increase aquatic metabolic rates, (3) increase biochemical oxygen demand, and (4) accelerate chemical processes (Curtis and others 1990). A 10° C increase in stream temperature from 5° C to 15° C can double the metabolic rate of fish and other aquatic organisms, and reduce the saturation concentration for dissolved oxygen by approximately 20 percent (National Council for Air and Stream Improvement 1994).

The National Council for Air and Stream Improvement (1994) compiled temperature effects of complete canopy removal from a variety of studies across the United States. Increases in summer temperatures ranged from about 2° to 12° C. In a study in central Pennsylvania (Lynch and others 1985), removal of riparian vegetation resulted in an increase in summer water temperatures of 5° to 11° C, while the retention of riparian vegetation minimized the increase to 1° to 2° C during the summer months. Beschta and others (1987) found that retaining canopy cover generally keeps temperature increases to less than 2° C. Hewlett and Fortson (1982) found that clearcutting in Georgia, while maintaining a partial buffer strip, increased average summer temperatures by 6.7° C. Swift and Messer (1971) found that clearcutting in Appalachian Mountain cove hardwoods increased average summer maximum temperatures by 2.8° to 3.3° C, while maintaining the overstory and simply cutting understory vegetation increased temperatures by only 0.3° C.

Scoles and others (1996) found that the average water temperature in unshaded pools in three small streams in southeast Oklahoma following harvest was 3.6° F higher at the water surface; temperatures at lower depths were unaffected. The streams were dry during the study except for a series of shallow pools (1 to 3 feet deep). Temperatures returned to normal downstream of the harvested area where groundwater inflow and streamside vegetation served to return temperatures to normal. Swift and Baker (1973) illustrate the cooling effects of shade strips contrasted with the stronger cooling by groundwater inflow.

Vowell (in press) reports that use of streamside buffers effectively maintained stream temperatures after clearcutting, intensive site preparation, and machine planting on four sites in northern Florida.

Only one study (Hewlett and Fortson 1982) reported major changes in stream temperatures after timber harvesting when riparian buffer strips were retained.

4.4 Dissolved Oxygen

Aquatic organisms need the oxygen dissolved in streamwater for metabolic activity. Dissolved oxygen concentrations can vary by stream because they depend on temperature and air pressure (elevation), as well as instream processes, including plant and animal respiration, oxygenation by means of gas exchange with the atmosphere, instream photosynthesis, and nutrient inputs. Dissolved oxygen concentrations vary diurnally due to instream plant and animal respiration. Concentrations of 8 mg/L are considered optimal for aquatic organism health (U.S. Environmental Protection Agency 1986, Chapman and McLeod 1987).

The impacts of forestry activities on dissolved oxygen levels in streambed sediments is less clear (National Council for Air and Stream Improvement 1994), but it seems likely that, in the absence of proper BMP implementation, increased fine sediment deposition may lead to decreased permeability of streambeds and thereby reduced intergravel oxygen concentrations (Everest and others 1987, Chapman and McLeod 1987). Reduced oxygen concentrations can lead to reduced viability of aquatic insects and fish eggs.

While there is limited research on this subject as it relates to forestry in the South, a few studies in Oregon (Hall and others 1987) and Quebec (Plamondon and others 1982) have documented that large inputs of fine litter to small, low-turbulence streams can deplete dissolved oxygen concentrations.

Vowell (in press) found that the use of BMPs in northern Florida adequately protected dissolved oxygen levels after clearcutting, intensive site preparation, and machine planting. Measurements before and after silvicultural treatments revealed no significant change in streamwater chemistry. Another study (Ensign and Mallin 2001) documenting the effects of forestry activities on dissolved oxygen is summarized in Section 4.7, Woody Wetlands.

4.5 Nutrients

Nutrient concentrations in streams flowing from forests vary widely depending on soil type and texture, parent material, climate, stand age, species composition, and atmospheric deposition. The U.S. Geological Survey conducted a national study of nutrient concentrations and yields in primarily undeveloped basins in an effort to more fully evaluate the effects of anthropogenic activities on water quality (Clark and others 2000). The majority of these basins were dominated by extensive forest cover and located primarily in wilderness areas, national and state parks, and national forests. The authors found that these basins produced the best water quality in the country. Concentrations of ammonia, nitrate, total nitrogen, orthophosphate, and total phosphorus rarely exceeded national water quality standards.

Few nutrients are lost from healthy forest ecosystems directly to stream channels. These systems are very efficient at recycling nutrients. Young forests rapidly soak up nutrients from the soil as they grow (Borman and Likens 1994, Scoles and others 1996). The sudden removal of vegetation through timber harvesting or insect infestation, however, can increase the nutrient transport to streams by increasing leaching and erosion (Scoles and others 1996). Most increases in stream nutrient levels occur in the first few years after harvesting. Stream concentrations rapidly decline back to pre-harvest levels as vegetation regrows. In contrast, Swank and others (1981) and Swank (1988) reported small but persistent—-as long as 20 years—-increases in nutrient concentrations following insect defoliation and forest cutting. The effects of increases in nutrient inputs are often diluted by increases in stormflow after harvest (Scoles and others 1996). However, increases in streamflow can also lead to increases in total loading of nutrients to downstream areas, particularly lakes and reservoirs. The impacts of increased nutrients due to fertilization are discussed in Section 4.8.4 of this Chapter.

The primary nutrients affecting ecological processes in streams and lakes are nitrogen (primarily as nitrate) and phosphorus (primarily as phosphate) (National Council for Air and Stream Improvement 1994). Increases in nitrogen and phosphorous concentrations can increase stream productivity, increase daily fluctuations in stream oxygen concentrations, and increase or decrease species diversity. Excessive amounts of nutrients may also stimulate algal blooms. Large blooms limit light penetration into the water column, increase turbidity, and increase biological oxygen demand, resulting in reduced dissolved oxygen levels. This process, termed eutrophication, drastically affects aquatic organisms.

According to Binkley and Brown (1993), most forest harvesting studies in the United States have documented increased concentrations of nitrate after harvest. With a few exceptions, these increases have remained well below the 10 mg/L drinking water standard (U.S. Environmental Protection Agency 1986). This standard is appropriate for waterbodies whose designated uses include municipal drinking water. However, aquatic communities respond to much lower levels of inorganic nitrogen. EPA is in the process of developing national nutrient standards for maintaining water quality that supports aquatic life and recreation as a designated use (U.S. Environmental Protection Agency 2000a).

One ecosytem region in the South that has been known to exceed the drinking water standard is high-elevation spruce-fir forest in the Southern Appalachian Mountains. Average nitrogen concentrations of 5 mg/L, with higher reported maximum values, occur in some streams in this area (Silsbee and Larson 1982). Factors possibly contributing to the elevated nitrogen concentrations include atmospheric nitrogen deposition and low nitrogen uptake rates due to the mature nature of these forests (Silsbee and Larson 1982). The Southern Appalachians receive relatively high rates of atmospheric nitrogen deposition compared to the rest of the region (Johnson and Lindberg 1992).

In a summary of several studies that considered the impacts of harvesting operations on nutrient inputs, Richter (2000) reports that streamwater nitrate nitrogen may increase up to 1 mg/L after harvest in the Appalachian Mountains (Swank 1988), the Atlantic Coastal Plain (Askew and Williams 1986, Riekirk 1983), the southern Piedmont (Hewlett and others 1984), and the Ouachita Mountains (Miller and others 1988).

In a study of the effectiveness of BMPs in northern Florida, Vowell (in press) reported that the State's BMPs adequately protected water quality. Streamwater chemistry, including total phosphorous, ammonia, nitrate and nitrite, showed no significant differences before or after harvest.

Scoles and others (1996) reported that nitrogen and phosphorus levels increased the first year after harvesting but returned to baseline conditions within 4 years.

Ammonia generally is not a problem since it is found in low concentrations due to its high adsorptivity and ready conversion to nitrate. Several studies found little or no change in ammonia concentrations (Martin and others 1984, Blackburn and Wood 1990). One study (Van Lear and others 1985) reported decreases in ammonia concentrations after tree harvests. Decreases were attributed to increased nitrification due to increased soil temperature and moisture following harvest.

Mostaghimi and others (1999) found that harvest and site-preparation activities without the use of BMPs significantly increased nutrient loss during storms in the Virginia Coastal Plain. Stormflow concentrations and loadings of nitrogen and phosphorus increased significantly. Where BMPs were not applied, harvesting increased nitrogen loading by a factor of 3.1, and site preparation activities increased it by a factor of 5.5. Use of BMPs mitigated these increases. In the absence of BMPs, total phosphorus in stormflow increased three- and four-fold following harvest and site preparation activities, respectively, as compared to preharvest conditions. Stormflow phosphorus loading decreased 45 percent on the BMP watershed following harvest and did not change significantly after site preparation.

4.6 Aquatic Habitat and Biota

Fish and invertebrates depend on a variety of stream physical characteristics including temperature, dissolved oxygen, turbidity, light, nutrients, sediment particle size distribution, and refuge opportunities. Chapter AQUA-5 contains a complete discussion on the range of aquatic habitats and species in the South. Most studies on the impacts of silvicultural activities on aquatic biota and habitat have been conducted in the Pacific Northwest and northwestern California, areas dominated by steep slopes, frequent landslides, erodible soils, and high precipitation levels. Under these conditions, forest practices can have a substantial impact on stream channel conditions if BMPs are not fully implemented and maintained over time.

Sullivan and others (1987) document several case studies in northern California that took place between 1950 and 1970 when several extreme storms after extensive logging resulted in substantial alterations to stream channel morphology. Streambeds were raised by as much as 4 meters, stream widths were doubled, stream channels were shifted, average particle size was increased, pools were filled in, riffles became less pronounced, summer flows were reduced, riparian vegetation was degraded, and stream banks were eroded. It was difficult to separate the contribution of harvesting impacts from the general storm effects, but fish populations declined over this period. In the 1982 National Fisheries Survey (Judy and others 1984), forestry activities were estimated to produce adverse effects on fish in about 7.5 percent of assessed river and stream miles, compared to 29.5 percent for agricultural land and 6.7 percent for urban areas.

Tebo (1955) studied the effects of early logging practices in steep mountainous watersheds on siltation and the impacts on bottom organisms in western North Carolina. There were no limitations on logging method and the logging operations were not supervised by the USDA Forest Service. Tebo (1955) compared the number and volume of bottom-dwelling organisms upstream of the harvested area to a site located below the mouth of the stream draining the logged watershed that received an accumulation of silt. The author found a statistically significantly larger population and higher volume of bottom-dwelling organisms at the control site upstream. After the removal of accumulated sediments and reduction in numbers of organisms due to flooding, the section of stream impacted by sedimentation still produced a slightly but statistically insignificantly lower number of organisms than the control section.

Vowell (in press) examined the effects of intensive forest management activities on aquatic habitat in northern Florida using a stream condition index (SCI) based on benthic macroinvertebrate sampling measures. Biological indicators such as this are believed to be more accurate measures of water quality than chemical indicators since the presence, or absence, and abundance of aquatic organisms, benthic macroinvertebrates in particular, better reflect the overall ecological health of waterbodies because they integrate pollutant stressors over time. Vowell also evaluated aquatic habitat using an average habitat assessment value based on a composite of physical stream attributes including substrate type and availability, water velocity, artificial channelization, habitat smothering, stream bank stability, riparian buffer width, and riparian buffer quality.

Vowell (in press) found no significant differences between pre- and post-treatment SCI values at any of the four sites, indicating no effect due to silvicultural activities. Average habitat assessment values were also within the "optimal" range both before and after treatments. The only notable differences found after treatment were changes in the score for water velocity and riparian zone width. The measured increase in water velocity was attributed to minor temporal variability rather than the treatment. Riparian zone widths after harvesting, while considered "marginal" from a scoring point of view, were still within the required width for primary streams. No change was recorded for habitat smothering or stream bank stability, two components of the habitat assessment considered especially sensitive to impacts from silvicultural activities and critical to maintaining macroinvertebrate population integrity.

Interestingly, some studies have actually documented increases in fish populations and fish size after logging (see Hall and Lantz 1969, Murphy and Hall 1981, Murphy and others 1981, Hawkins and others 1983). These increases are generally attributed to alterations in the foodweb (National Council for Air and Stream Improvement 1994). For example, increased light penetration or nutrient concentrations may lead to increases in primary productivity that may increase herbivore populations. Slight increases in stream temperature can actually favor fish growth and increase survival of young fish, particularly in northern latitudes or high-elevation streams (Holtby 1988)

4.7 Woody Wetlands

Forested wetlands are important for their ability to transform inorganic nutrients into organic form, as well as filter out sediment and particulate matter (Lockaby and others 1997). Forested wetlands were considered unproductive up to the 1950s, when many large pine plantations were established on drained forested wetland sites in the Lower Coastal Plain of the South (Wu and others 1999). Forested wetlands are characterized by high seasonal water tables and soil surface waterlogging due to flat topography and poor soil drainage. A brief discussion of the impacts of silvicultural activities on forested wetlands is included below; however, a complete discussion of forested wetland characteristics and potential impacts from various land-use activities, including silviculture, is included in Chapter AQUA-2.

The primary silvicultural activities potentially affecting important wetland functions are site drainage and the operation of heavy equipment on wetland soils, usually during site preparation. Site drainage improves access, provides for soil aeration, and increases seedling survival and growth (Segal and others 1987). Site preparation practices such as mole-plowing and bedding are among the most prominent silvicultural practices in the South (Wu and others 1999). Mole-plowing uses a deep plow to create a channel in poorly drained soils to improve site drainage. Bedding is a common practice that elevates planted trees on beds above the surface of the water table. Minor drainage is often needed to remove excess surface water to permit heavy equipment to be operated without causing extensive soil compaction and rutting (Shepard 1994).

In contrast to upland forests, surface water flow rates are low in wetlands, which typically have little topographic relief, and therefore have less energy available to export sediment. In a review of literature on water quality in forested wetlands, Shepard (1994) found that silvicultural activities generally resulted in water-quality impacts but the impacts were typically small and short-lived. Impacts were greater in upland wetlands where relief is greater and soils are shallower than in lowland wetland forests. Impacts on all common wetland types have not been investigated. In particular, there is very little published information available on the impacts from bottomland hardwood silviculture on water quality. Shepard (1994) concludes that silvicultural activities "do not constitute a permanent threat to the ability of wetlands to maintain or improve water quality."

Wu and others (1999) examined the effects of clearcutting in the wet and dry seasons and site preparation activities (bedding and mole-plowing plus bedding) on groundwater levels. The authors found that water tables rose in response to forest removal, with the greatest increases occurring after wet-weather logging. The larger increase associated with wet-weather harvesting was likely due to deeper rutting and greater soil disturbance. No significant differences in groundwater levels were found during the dormant season, indicating that the removal of transpiring vegetation was primarily responsible for the increase in water table levels (Wu and others 1999).

The same study found that site-preparation techniques ameliorated harvest-related elevated water tables by improving site drainage. Bedding reduced groundwater level by up to 22 cm compared to nonbedded sites. Mole-plowing plus bedding had a similar effect on groundwater levels as bedding alone. The recovery of site hydrology was fastest on sites that had been the least disturbed -- harvested during dry weather and bedded only. Site hydrology recovered within 2 years of stand establishment (Wu and others 1999).

Miwa and others (1999) also found that wet-weather harvesting had a significantly larger impact on site hydrology than did dry-weather treatment.

Riekerk (1985) conducted a comparative watershed study in the poorly drained pine flatwoods of northern Florida. One watershed was clearcut with minimum disturbance and site preparation (manual shortwood harvesting, slash chopping, soil bedding, and machine planting). The second watershed was clearcut with maximum disturbance and site preparation (machine tree-length harvesting, slash burning, windrowing, soil bedding, and machine planting). The third watershed was an undisturbed control. Runoff increased 2.5-fold on the minimum treatment watershed and increased 4.2-fold on the maximum treatment watershed. There was a statistically significant increase in the level of suspended sediment (14 ppm on average) proportional to disturbance, but the absolute levels were low. Significant increases over the control remained for 4 years after both treatments (Riekerk 1985).

Ensign and Mallin (2001) studied the water-quality impacts of clearcutting 130 acres of riparian and seasonally flooded forest in the Coastal Plain of North Carolina. The authors found short-term increases in stream turbidity reaching 111 nephalometric turbidity units (NTU), well above the North Carolina State standard of 50 NTU, but the average increase was not statistically significant. However, compared with an unlogged control stream, suspended sediment concentrations were significantly increased for several months after the clearcut. In addition, statistically significant post-logging increases were reported for both total nitrogen and total phosphorus compared to a nearby control stream.

In aquatic habitats, Ensign and Mallin (2001) found significant decreases in dissolved oxygen that approached anoxia on several occasions after timber harvest. The decreases were attributed to stream algal blooms that formed periodically for two summers after clearcutting. The blooms occurred from a combination of increased nutrient inputs and possibly increased direct solar radiation on surface water. The formation of algal blooms, followed by death and decomposition, created high biochemical oxygen demands leading to decreased dissolved oxygen levels.

Another biotic parameter of interest in streams with human recreation as a designated use is microbial pathogens. Ensign and Mallin (2001) found greatly increased fecal coliform bacterial concentrations in streams following clearcutting. This increase may have occurred due to runoff of pathogens from nearby large-scale swine production facilities, or from the land disturbance itself (Ensign and Mallin 2001).

Lebo and Herrman (1998) examined outflow characteristics in a low-level pocosin with artificial drainage in a 1,161-acre watershed and found that sediment export from the watershed increased nearly 350 percent (4.1 lbs/acre to 14.3 lbs/acre) during a 3-year period that included harvest and site preparation activities. Minor increases in nitrogen concentrations in streamwater were detected after harvest. These concentrations were typically less than the average value for the control stand. Increases in phosphorus concentrations were more prolonged than for nitrogen, but they decreased to preharvest levels after 3 years.

4.8 Management Intensity

This section describes the gradient of potential water quality impacts across a variety of silvicultural management techniques. The activities discussed include: (1) the harvesting method (single-tree selection, group selection, and clearcutting); (2) the degree of mechanization used in felling and collecting logs (hand felling, feller-bunchers, and cable yarders); and (3) the site preparation method (windrowing, shearing, disking, prescribed burning, and use of fertilizers and herbicides). Other aspects of timber management associated with management intensity but not related to site disturbance and sedimentation, such as the conversion of hardwood and natural pine stands to pine plantations, are covered in the final section of this Chapter. A more thorough discussion of forest operation technologies, including various site preparation techniques and their impacts on the environment, is included in Chapter TIMBR-3.

In general, as management intensity increases so does the level of site disturbance. Similarly, the greater the site disturbance, the greater the nonpoint-source impacts, particularly increased erosion and potential for sediment delivery into streams (Riekerk 1985). For example, in the poorly drained pine flatwoods of northern Florida, Riekerk (1985) found increases in total runoff, pH, suspended sediment, and potassium and calcium concentrations proportional to site disturbance in the year after harvest.

4.8.1 Effects of Harvest Method

It is widely acknowledged that the majority of effects from silvicultural activities can be attributed to operation of heavy machinery on roads and skid trails near waterbodies. Rice and Wallis (1962) found no detectable change in stream channel conditions following harvest other than impacts directly resulting from logging equipment and logging debris. Physical alterations included stream channel scouring or filling by bulldozers, slash and debris in channel crossings, and diversion of water down logging roads at stream crossings and road cuts. The diversions caused severe gullying.

McMinn (1984) compared a skidder logging system and a cable yarder for their relative effects on soil disturbance. With the cable yarder, 99 percent of the soil remained undisturbed (the original litter still covered the mineral soil), while the amount of soil remaining undisturbed after logging by skidder was only 63 percent. Currently, cable yarding is primarily limited to the steepest slopes in the Appalachian Mountains and is otherwise rarely used in the South.

Other studies have demonstrated that the intensity of harvest, depending on the silvicultural prescription, may increase concentrations and loadings of sediment during storms. In watershed research studies in Arkansas and Oklahoma, Scoles and others (1996) found that soil loss increased with harvest intensity (clearcutingt versus selection harvesting). Site preparation activities consisted of crushing and burning residual vegetation. No special erosion control measures were applied. In both studies, statistically significant increases in annual soil loss were found in the first year after clearcutting compared to selectively harvested and control sites. Annual soil losses averaged 211 lbs/acre and 251 lbs/acre on clearcut watersheds in Arkansas and Oklahoma, respectively.

Research conducted by Beasley and Granillo (1985) demonstrated that selective cutting generated lower water yields and sediment yields than did clearcutting. Selective cutting resulted in sediment yields 2.5 to 20 times less and water yields 1.3 to 2.6 times less than those resulting from clearcutting.

Eschner and Larmoyeux (1963) completed a study that compared the water-quality impacts from four harvesting methods: (1) commercial clearcut, (2) intensive selection (trees over 5 inches diameter breast height (DBH) were cut), (3) extensive selection (trees over 11 inches DBH were cut), and (4) diameter limit (trees over 17 inches DBH were cut). However, each of these harvest methods was combined with varying road designs, to determine their overall effectiveness in protecting water quality. It was concluded that the amount of trees removed, or harvesting method, was not the primary factor affecting water quality, as measured by turbidity. Water-quality impacts were shown to be related to the care taken in logging and planning skid roads. The extensive selection method, combined with some nonpoint-source controls (20 percent road grade limits, no skidding in streams, water bars on skid roads), produced higher maximum levels of turbidity than did intensive selection (210 Turbidity Units and 25 Turbidity Units, respectively) with additional control practices (10 percent road grade limits; skid trails located away from streams). Harvesting by diameter limit without any restrictions on road grades or stream restrictions increased maximum turbidity by 200 times over intensive selection (5,200 Turbidity Units and 25 Turbidity Units, respectively). Commercial clearcutting with no controls increased maximum turbidity by over three orders of magnitude compared to harvesting by diameter limit (56,000 Turbidity Units and 25 Turbidity Units, respectively).

4.8.2 Effects of Site Preparation

Shearing, disking, drum-chopping, or root-raking a site with large tractors may heavily disturb the soil over large areas and has a high potential to deteriorate water quality (Beasley 1979). Site preparation techniques that remove vegetation and litter cover, compact the soil, expose or disturb the mineral soil, and increase stormflows due to decreased infiltration and percolation, all can contribute to increases in sediment loads (Golden and others 1984). However, erosion rates typically decrease as vegetative cover grows back. Prescribed burning and application of herbicides and fertilizers also have potential negative effects on water quality. These activities are discussed separately in sections that follow.

Shearing, which exposes large amounts of bare soil while removing logging debris, and windrowing resulted in higher levels of soil loss in the Texas Coastal Plain and Athens Plateau (Scoles and others 1996). Shearing also reduced the soil's ability to absorb water in the Texas study. Douglass (1977) found that total soil loss from sites that had been cleared was approximately 580 pounds of soil per inch of runoff. However, runoff from sites that were both cleared and disked was twice that from sites that had been cleared only.

Blackburn and Wood (1990) reported that harvesting and shearing a watershed in east Texas increased phosphate and total phosphorus concentrations in the year after harvest, while harvesting and chopping had no effect on phosphate and total phosphorus concentrations.

As described previously, Wu and others (1999) determined that site preparation activities (bedding and mole-plowing plus bedding) reduced water table levels significantly in forested wetlands.

4.8.3 Effects of Prescribed Fire

Prescribed fire can impact water quality by heating the soil, and killing soil organisms, thereby altering nutrient transformation rates and bioavailability. These impacts depend on the severity and intensity of the fire. Prescribed burning of slash can increase erosion and sediment delivery to streams by eliminating protective cover and altering soil properties (Megahan 1980). The degree of erosion after a prescribed burn depends on soil erodibility; slope; precipitation timing, volume and intensity; fire severity; cover remaining on the soil; and speed of revegetation. Swift and others (1993) found erosion after burning to be spotty and did not leave the treated site or reach stream channels. The prescription for this burn, however, was to maintain a low fire intensity and avoid consuming the compacted litter or organic layers. Burning may also increase stormflow in areas where all vegetation is killed. Such increases are partially attributable to decreased evapotranspiration rates and reduced canopy interception of precipitation. Erosion resulting from prescribed burning is generally less than that resulting from roads and skid trails and from site preparation that causes intense soil disturbance (Golden and others 1984).

Knoepp and Swank (1993) found that clearcutting and burning increased streamwater nitrate concentrations from less than 0.01 mg/L to a maximum of 0.075 mg/L. This small increase was associated with a slight increase in nitrogen transformations and little movement of inorganic nitrogen off the site (Knoepp and Swank 1993). Concentrations returned to pretreatment levels within 9 months after burning.

In a paired watershed study, Van Lear and others (1985) examined soil and nutrient export in ephemeral streamflow after three low-intensity prescribed fires prior to harvest on the Clemson Experimental Forest in the Upper Piedmont of South Carolina. Minor increases in stormflow and nutrient and sediment concentrations in the water were identified after low-intensity prescribed fires. It was suggested that erosion and sedimentation from plowed fire lines accounted for the majority of sediment from all watersheds. Following the prescribed fires, the overstory in the burned watersheds was harvested, and runoff, sediment and nutrient export were monitored for 3 years after harvest. Sediment levels were elevated after harvest, but the magnitude and duration of these effects were considerably less than from other studies (Douglass and Goodwin 1980, Fox and others 1983, Hewlett 1979) that utilized mechanical site preparations techniques, instead of prescribed burning (Van Lear and others 1985).

Landsberg and Tiedemann (2000) thoroughly reviewed the effects of wildfires and fire management on water quality. The following specific management measures were identified as ways to reduce the magnitude of the effects of fire on water quality: (1) limit fire severity, (2) avoid burning on steep slopes, and (3) limit burning on sandy or water-repellent soils.

4.8.4 Fertilizers, Pesticides, and Herbicides

Although fertilizer application is uncommon in hardwood forests in the East, forest fertilization is routine--and possibly increasing (Dubois and others 1999)--on many intensively managed pine plantations in the South (Shepard 1994). A brief discussion of the use of fertilizers and pesticides (herbicides and insecticides) in forest operations is included in Chapter TIMBR-3. In a periodic survey of the cost of forest practices, Dubois and others (1999) report that the number of fertilized acres increased between 1996 and 1998. Few studies have looked at the impacts of this practice on water quality (Shepard 1994). Studies typically show that forest fertilization is not a problem; most studies have shown that nutrient increases are too small to degrade water quality (Binkley and Brown 1993, Fisher and Binkley 2000). Many forest streams are nutrient limited, so the application of fertilizers has a greater potential for impacts in nutrient-poor aquatic ecosystems.

Fertilizers, pesticides, and herbicides reach streams either directly through aerial or hand application, or indirectly by surface runoff and subsurface flow. BMPs typically restrict application to nonriparian zones. However, in practice, riparian zones are difficult to avoid in aerial applications. The effects of fertilizer application on aquatic ecosystems are the same as described for nutrients in Section 4.6 of this Chapter.

Pesticides can have both direct and indirect effects on ecological processes. Aquatic organisms can be affected through direct exposure to pesticides in the streamwater or through ingestion. There have been too few studies on the impacts of insecticides to make generalizations about the impacts on fish populations. Some 1- to 2-year studies (Reed 1966) have concluded that short-term reductions in insect populations--an important food source for fish--may occur. Insect communities should recover within a few years due to their short life cycles (National Council for Air and Stream Improvement 1994).

Herbicides can impact aquatic communities directly through increased organic matter inputs and indirectly through other effects on riparian vegetation. These secondary impacts can include changes in physical properties of streams, such as increases in water temperature and sedimentation, due to loss of riparian vegetation. Other secondary impacts to stream properties can result from changes in riparian vegetation, including increased nitrate inputs, decreased slope stability, and altered food web structure in streams. No critical indirect effects have been documented for normal forest use of herbicides (National Council for Air and Stream Improvement 1994).

In a literature review on forest fertilization with nitrogen and phosphorus and water quality, Binkley and others (1999) found that without the use of BMPs, short-lived elevated nitrate and phosphorus concentrations were often found in receiving waters, but that national drinking-water-quality standards (for nitrogen) and/or suggested criteria (for phosphorus) were rarely exceeded. No studies were identified that reported adverse affects on aquatic biota.

The effects of fertilizer application on water quality were studied in three North Carolina plantations (Campbell 1989). Fertilization temporarily elevated levels of ammonium, total nitrogen, total phosphate, orthophosphate, and urea in streams draining plantations. Concentrations returned to pretreatment levels within 3 weeks. Net exports were small compared to the total amount of fertilizer applied: net export of total Kjeldahl nitrogen was 0.3 percent of total nitrogen applied, net export of ammonium was 0.02 percent of total nitrogen applied, and net export of urea was 0.03 percent of total applied urea. Several other studies reported similar results (Herrmann and White 1983, Fromm 1992).

Segal and others (1987) studied the effects on water quality of applying fertilizer and herbicide in a pine flatwood in eastern South Carolina. They identified a strong pulse of nutrient concentrations in July and attributed this to higher mineralization rates of forest floor litter and higher soil temperatures after clearcutting. Nutrient concentrations in groundwater did not appear to be outside the range of natural seasonal nutrient dynamics. Furthermore, groundwater quality did not appear to be negatively affected. All nutrient levels returned to pretreatment levels within 200 days after fertilizer application.

4.9 Hardwood Conversion to Planted Pine

Swank and Vose (1994) summarized over 40 years of research on changes in water yield and timing of streamflow, and over 20 years of stream chemistry data after conversion from hardwood forests to eastern white pine plantations. Significant decreases in water yield (up to 25 percent) were attributed to greater leaf area index throughout the year and, consequently, greater interception loss in the dormant season, plus greater transpiration loss in the early spring and late fall, and on warm winter days. The magnitudes of high and low flows were reduced by 33 to 60 percent. Streamwater solute concentrations remained similar on the pine and hardwood watersheds. Net accumulations of calcium, magnesium, potassium, and sodium increased by 1.1 lbs/acre to 3.9 lbs/acre on the pine watershed. Decreases in water yield resulting from extensive hardwood conversion to planted pine could potentially impact the availability of future water yields for municipal water supplies.

In a series of papers on timber harvesting in a Carolina bay, Askew and Williams (Askew and Williams 1984, 1986, and Williams and Askew 1988) concluded that pine plantations could be established without harming water quality. Askew and Williams (1984) quantified suspended sediment in drainage waters from active logging sites, site-prepared areas, 3- to 15-year old plantations, and main ditches in a 2,388 ha Carolina bay in South Carolina. Suspended sediment in new secondary ditches was significantly greater than in native streams draining an undisturbed hardwood stand. Water in the main ditch near the discharge point averaged 16.4 mg/L, compared to 2.5 mg/L in the undisturbed hardwood forest. Ditch contributions to suspended sediment concentrations were transient, culminating within 2 years of installation.

5 Discussion and Conclusions

The effect of silvicultural activities on water quality is often contentiously debated. Forestry operations have been identified as nonpoint sources of pollution to waterbodies draining forestland. Silvicultural activities have the potential to increase sedimentation and alter stream-channel conditions (National Council for Air and Stream Improvement 1994). Impacts from these activities are site-specific, varying across the South. Effects depend on elevation, slope, and the rate at which vegetation recovers following harvest. However, in general, if BMPs are properly designed and implemented, the adverse effects of forestry activities on hydrologic response, sediment delivery, stream temperature, dissolved oxygen, and concentrations of nutrients and pesticides can be minimized.

One of the objectives of sustainable forest management is to ensure that silvicultural activities are conducted without significant nonpoint-source pollution of streams and coastal areas. This Chapter identified the primary and secondary impacts of silvicultural operations. The following specific management measures should be considered by all forest managers as they develop comprehensive forest management plans. The effectiveness of these management measures to mitigate water quality impacts is discussed exclusively in Chapter AQUA-4.

Planning of the timber harvest to ensure water-quality protection will minimize nonpoint-source pollution and increase operational efficiency (Golden and others 1984). Streamside management areas of sufficient width and extent are crucial because they can greatly reduce pollutant delivery. Identification and avoidance of high-hazard areas can greatly reduce the risk of landslides and mass erosion. Careful planning of roads and skid trails will reduce the amount of land disturbed by them, thereby reducing erosion and sedimentation (Rothwell 1978). Proper design of drainage systems and stream crossings can prevent system destruction by storms, thereby preventing severe erosion, sedimentation, and channel scouring (Swift 1984a).

Road system planning is a critical part of preharvest planning. Good road location and design can greatly reduce the sources and transport of sediment. Road systems should generally be designed to minimize the number of road miles/acres, the size and number of landings, the number of skid trail miles, and the number of watercourse crossings, especially in sensitive watersheds. Timing operations to take advantage of favorable seasons or conditions and avoiding wet seasons prone to severe erosion or spawning periods for fish reduce impacts to water quality and aquatic organisms (Hynson and others 1982). Drainage problems can be minimized when locating roads by avoiding clay beds, seeps, springs, concave slopes, ravines, draws, and stream bottoms (Rothwell 1978). Stringer and Thompson (2000) attribute the limited use of topographic maps by loggers and silvicultural operators for many impacts to water quality.

Potential water-quality and habitat impacts should also be considered when selecting the silvicultural harvest and yarding systems. It may appear to be beneficial to water quality to use uneven-aged silvicultural systems because they disturb less ground and remove less of the canopy than clearcuts. These factors, however, should be weighed against the possible adverse effects of harvesting more acres selectively to yield equivalent timber volumes. Such harvesting may require more miles of roads and more frequent re-entry into timber stands, which can increase sediment generation. Whichever silvicultural system is selected, preharvest planning should address how harvested areas will be regenerated to prevent erosion and potential impact to waterbodies.

Cumulative effects to water quality from forest practices are not well documented (Neary and others 1989, Reid 1993, Vowell in press). They are related to several processes -- onsite mass erosion, onsite surface erosion, pollutant transport and routing, and receiving water effects (Sidle 1989). Cumulative effects are influenced by forest management activities, natural ecosystem processes, and the distribution of other land uses. Timber harvesting, road construction, and chemical use may directly affect onsite delivery of nonpoint-source pollutants as well as contribute to existing cumulative impairments of water quality. The most effective road system results from planning to serve an entire basin, rather than arbitrarily constructing individual roads to serve short-term needs (Swift 1985).

On watersheds where cumulative effects are known to be a problem, the potential for additional water-quality impairments should be taken into account during preharvest planning. Information from previously conducted watershed assessments should be considered. These types of assessments, generally conducted by State or Federal agencies, may indicate water-quality impairments in watersheds of concern caused by types of pollutants unrelated to forestry activities. However, if existing assessments attribute a water-quality problem to the types of pollutants potentially generated by the planned forestry activity, then the problem should be considered during the planning process. If additional contributions to this impairment are likely to occur, planned activities may have to be adjusted or additional mitigation measures may have to be implemented. Alterations may include selection of harvest units with low sedimentation risk, such as flat ridges or broad valleys; postponement of harvesting until existing erosion sources are stabilized; and selection of limited harvest areas using existing roads.

6 Needs for Additional Research

The nonpoint-source literature is heavily weighted to hydrologic response, sedimentation, and nutrients, the primary silvicultural impacts. Relatively little research has been completed for southern aquatic ecosystems related to channel morphology, dissolved oxygen, pH, woody debris loading, aquatic habitat and biota, hardwood conversion, municipal water supplies, nutrient impacts to lakes, and prescribed fire. Specifically, there is further need to investigate comprehensive biotic impacts from silviculture, including phytoplankton and macroalgal blooms, food-chain impacts, and potential increased microbial pathogen runoff.

While the available research indicates that individual forestry operations do not contribute significantly to water-quality impairment when BMPs are effectively implemented and monitored, additional research is necessary to assess the long-term cumulative nonpoint-source impacts of silvicultural activities on water quality and overall watershed health. Given the nature of land ownership patterns in the South, this additional research should be conducted by public-private partnerships, with cooperation from forest industry, government agencies, academia, and other interested groups.

The National Council for Air and Stream Improvement (1994) identified the following research needs: (1) development of and testing of more stringent BMPs for some locations, (2) improvement of the capability to effectively evaluate risk, and (3) focus of future research on erosion, sedimentation, and effects on stream channels and fish habitat. Three major areas warrant additional attention: high-nitrate systems, operational-scale assessments of BMP effectiveness (California State Water Resources Control Board 1987, Knopp and others 1987, and Harvey and others 1988), and the cumulative effects of management practices within basins.

7 Acknowledgments

The authors thank the numerous members of the general public who provided relevant forestry studies and water quality references. We especially thank Lloyd Swift, who served as the primary panel expert in February 2000. He provided invaluable assistance in developing the plan of work and provided a significant number of literature citations from research conducted at the USDA Forest Service Coweeta Hydrologic Laboratory in Otto, NC.

8 Literature Cited

Askew, G.R. and T.M. Williams. 1984. Sediment concentrations from intensively prepared wetland sites. Southern Journal of Applied Forestry. 8:152-157.

Askew, G.R. and T.M. Williams. 1986. Water quality changes due to site conversion in coastal South Carolina. Southern Journal of Applied Forestry. 10:134-136.

Beasley, R.S. 1979. Intensive site preparation and sediment losses on steep watersheds in the Gulf Coastal Plain. Soil Science Society of America Journal. 24(2):361-366.

Beasley, R.S. and A.B. Granillo. 1985. Water yields and sediment losses from chemical and mechanical site preparation in southwest Arkansas. pp. 106-116. In: B.G. Blackmon, ed., Proceedings from Forestry and Water Quality: a Mid-South Symposium, Little Rock, AR: University of Arkansas at Monticello.

Beasley, R.S., and A.B. Granillo. 1988. Sediment and water yields from managed forests on flat coastal plain soils. Water Resources Bulletin, 24(2): 361-366.

Beasley, R.S., E.L. Miller, and S.C. Gough. 1984. Forest road erosion in the Ouachita Mountain. pp. 203-212. In: Peters, P.A. and J. Luchok, eds., Proceedings from Mountain Logging Symposium, Charleston, WV: West Virginia University.

Beschta, R.L., R.E. Bilby, G.W. Brown, L.B. Holtby, and T.D. Hofstra. 1987. Stream temperature and aquatic habitat: fisheries and forestry interactions. pp. 191-232. In: Salo, E.O. and T.W. Cundy, eds., Streamside management: forestry and fishery interactions. Seattle, WA: College of Forest Resources Contribution #57, University of Washington..

Bethea, J.M. 1985. Perspectives on nonpoint source pollution control: silviculture. p. 13. In: Proceedings from Perspectives on Nonpoint Source Pollution, Kansas City, Missouri. Washington, D.C: U.S. Environmental Protection Agency, Office of Water Regulations and Standards.

Binkley, D. and T.C. Brown. 1993. Forest practices as nonpoint sources of pollution in North America. Water Resources Bulletin. 29(5):729-740.

Binkley, D., D.H. Burnham, and H.L. Allen. 1999. Water quality impacts of forest fertilization with nitrogen and phosphorous. Forest Ecology and Management. 121:191-213.

Blackburn, W.H. and J.C. Wood. 1990. Nutrient export in stormflow following forest harvest and site-preparation in East Texas. Journal of Environmental Quality. 19:402-408.

Blackburn, W.H, J.C. Wood, and M.G. DeHaven. 1986. Stormflow and sediment losses from site-prepared forestland in East Texas. Water Resources Research. 22(5):776-784.

Bosch, J.M. and J.D. Hewlett. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. Journal of Hydrology. 55:3-23.

Bormann, F.H. and G.E. Likens. 1994. Pattern and process in a forested ecosystem: disturbance, development, and the steady state based on the Hubbard Brook ecosystem study. New York, NY: Springer-Verlag. 253 pp.

Brown, G.W. 1972. Logging and water quality in the Pacific Northwest. Pp. 330-334. In: Proceedings from Watersheds in Transition Symposium. Urbana, IL: American Water Resources Association.

Brown, G.W. 1985. Controlling nonpoint source pollution from silvicultural operations: What we know and don't know. Pp. 332-333. In Proceedings of a National Conference, Perspectives on Nonpoint Source Pollution, Kansas City, Missouri. Washington, D.C: U.S. Environmental Protection Agency.

Brown, T. C. and D. Binkley. 1993. Forest practices as nonpoint sources of pollution in North America. Water Resources Bulletin. 29(5):729-740.

Brown, T. C. and D. Binkley. 1994. Effect of management on water quality in North American forests. General Technical Report RM-248. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 27 pp.

California State Water Resources Control Board. 1987. Final report of the Forest Practices Rules Assessment Team. Sacramento, CA: California State Water Resources Control Board.

Campbell, R.G. 1989. Water quality mid-year report. Weyerhaeuser Research and Development Report. New Bern, NC: New Bern Forestry Research Station.

Chapman, D.W. and K.P. McLeod. 1987. Development of criteria for fine sediment in the Northern Rockies ecoregion. U.S. Environmental Protection Agency Water Division Report, 910/9-87-162. Seattle, WA.

Clark, G.M., D.K. Mueller, and M.A. Mast. 2000. Nutrient concentrations and yields in undeveloped basins of the United States. Journal of the American Water Resources Association. 36(4):849-860.

Curtis, J.G., D.W. Pelren, D.B. George, V.D. Adams, and J.B. Layzer. 1990. Effectiveness of best management practices in preventing degradation of streams caused by silvicultural activities in Pickett State Forest, Tennessee. Nashville, TN: Tennessee Technological University, Center for the Management, Utilization and Protection of Water Resources.

Dils, R.E. 1957. A guide to the Coweeta Hydrologic Laboratory. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 40 p.

Dissmeyer, G.E., ed. 2000. Drinking water from forests and grasslands: a synthesis of the scientific literature. Gen. Tech. Rep. SRS-039, Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 246 pp.

Douglass, J.E. 1977. Site preparation alternatives: quantifying their effects on soil and water resources. In: Proceedings of Site Preparation Workshop, East. Raleigh, North Carolina: U.S. Department of Agriculture Forest Service,.

Douglass, J.E. and O.C. Goodwin. 1980. Runoff and soil erosion from forest site preparation practices. pp. 92-151. In: U.S. Forestry and Water Quality: What Course in the 80s? Proceedings of the Water Pollution Control Federation Seminar, Richmond, VA.

Dubois, M.R., K. McNabb, and T.J. Straka. 1999. Costs and cost trends for forestry practices in the South. Forest Landowner, 58(2):3-8.

Ensign, S.H. and M.A. Mallin. 2001. Stream water quality changes following timber harvest in a coastal plain swamp forest. Water Research. 35(14):3381-3390.

Eschner, A.R. and J. Larmoyeux. 1963. Logging and trout: four experimental forest practices and their effect on water quality. Progress in Fish Culture. 25:59-67.

Everest, F.H., R.L. Beschta, J.C. Scriverner, K.V. Koski, J.R. Sedell, and C.F. Cederhom. 1987. Fine sediment and salmonid production: a paradox. pp. 98-142. In: Salo, E.O. and T.W. Cundy, eds., Streamside management: forestry and fishery interactions. Seattle, WA College of Forest Resources Contribution #57. University of Washington. :

Fisher, R.F. and D. Binkley. 2000. Forest soils. New York, NY: J. Wiley,

Fox, T.R., J.A. Burger, and R.E. Kreh. 1983. Impact of site preparation on nutrient dynamics and stream water quality on a Piedmont site. p. 207. In: Agronomy Abstracts, Madison, WI: American Society of Agronomy.

Fromm, J.H. 1992. Jones 5 plantation fertilizer runoff monitoring. Weyerhaeuser Research and Development Report, New Bern, NC: New Bern Forestry Research Station.

Golden, M.S., C.L. Tuttle, J.S. Kush, and J.M. Bradley. 1984. Forestry activities and water quality in Alabama: Effects, recommended practices, and an erosion-classified system. Auburn, AL:Auburn University Agricultural Experiment Station, Bulletin 555.

Hall, J.D. and R.L. Lantz. 1969. Effects of logging on the habitat of coho salmon and cutthroat trout in coastal streams. pp. 355-375. In: Symposium on salmon and trout in streams, H.R. MacMillan Lectures in Fisheries, Vancouver, BC: University of British Columbia..

Hall, J.D., G.W. Brown, and R.L. Lantz. 1987. The Alsea watershed study: a retrospective. pp. 399-416. In: Salo, E.O. and T.W. Cundy, eds., Streamside management: forestry and fishery interactions. Contribution #57 College of Forest Resources, Seattle, WA: University of Washington,.

Harvey, G., S. Hess, L. Jones, D. Krutina, D. McGreer, W. Reid, D. Stockton, and J. Thornton. 1988. Final report: forest practices water quality audit—1988. Boise, ID: Idaho Department. of Health and Welfare.

Hawkins, C.P., M.L. Murphy, N.H. Anderson, and M.A. Wilzbach. 1983. Density of fish and salamanders in relation to riparian canopy and physical habitat in streams of the northwestern United States. Canadian Journal of Fisheries and Aquatic Science. 40:1173-1185.

Helvey, J.D. and J.N. Kochenderfer. 1987. Effects of limestone gravel on selected chemical properties of road runoff and streamflow. Northern Journal of Applied Forestry. 4(1):23-25.

Herrmann, R. and W.M. White. 1983. Jones 5 plantation fertilizer runoff monitoring.

Weyerhaeuser Research and Development Report, New Bern, NC: New Bern Forestry Research Station.

Hewlett, J.D. 1961. Soil moisture as a source of base flow from steep mountain watersheds. Stn. Pap. 132. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 11 pp.

Hewlett, J.D. 1979. Forest water quality: An experiment in harvesting and regenerating Piedmont forests. Georgia Forestry Research Paper. Athens, GA.University: of Georgia.

Hewlett, J.D. and J.C. Fortson. 1982. Stream temperature under an inadequate buffer strip in the southeast Piedmont. Water Resources Bulletin. 18:983-988.

Hewlett, J.D. and J.D. Helvey. 1970. Effects of forest clear-felling on the storm hydrograph. Water Resources Research. 6:768-782.

Hewlett, J.D. and A.R. Hibbert. 1966. Factors affecting the response of small watersheds to precipitation in humid areas. In: Proceedings of a National Science Foundation advanced science seminar, international symposium of forest hydrology; 1965 August 29 - September 10; University Park, PA. New York: Pergamon Press: 275-290.

Hewlett, J.D., H.E. Post, and R. Doss. 1984. Effect of clear-cut silviculture on dissolved ion export and water yield in the Piedmont. Water Resources Research. 20:1030-1038.

Hibbert, A.R. 1966. Forest treatment effects on water yield. In: Proceedings of a National Science Foundation advanced science seminar, international symposium of forest hydrology; 1965 August 29 - September 10; University Park, PA. New York: Pergamon Press: 275-290.

Holtby, L.B. 1988. The effects of logging on stream temperatures at Carnation Creek. pp. 118-122. In: Chamberlin, T.W., ed., Proceedings of the workshop: Applying 15 years of Carnation Creek results. Nanaimo, BC: Pacific Biological Station.

Hynson, J., P. Adamus, S. Tibbetts, and R. Darnell. 1982. Handbook for protection of fish and wildlife from construction of farm and forest roads. FWS/OBS-82/18. Washington DC: U.S. Fish and Wildlife Service.

Johnson, D. and S. Lindberg. 1992. Atmospheric deposition and nutrient cycling in forest ecosystems. New York, NY: Springer-Verlag.

Judy, R.D., Jr., P.N. Seeley, T.M. Murray, S.C. Svirsky, M.R. Whitworth, and L.S. Ischinger. 1984. 1982 National fisheries survey. Volume 1, Technical report: Initial findings. FSW/OBS-84/06. Washington, DC: U.S. Fish and Wildlife Service.

Knoepp, J.D. and W.T. Swank. 1993. Site preparation burning to improve southern Appalachian pine-hardwood stands: nitrogen responses in soil, soil water, and streams. Canadian Journal of Forest Resources. 23:2263-2270.

Knopp, C.M., M.E. Smith, J. Barnes, B. Roath, and M.J. Furniss. 1987. Monitoring effectiveness of best management practices on National Forest lands. Pp. 48-54. In: Proc. Of the California Watershed Management Conference (R.Z. Callaham, J.J. DeVries, eds.). Report #11. Berkeley, CA: Wildland Resources Center, University of California.

Landsberg, J.D. and A.R. Tiedemann. 2000. Fire Management. pp. 124-138. In: Dissmeyer, G.E., ed. Drinking water from forests and grasslands: a synthesis of the scientific literature. Gen. Tech. Rep. SRS-039. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station.

Lebo, M.E. and R.B. Herrman. 1998. Harvest impacts on forest outflow in coastal North Carolina. Journal of Environmental Quality. 27(6):1382-1395.

Lieberman, J.S. and M.D. Hoover. 1948. The effect of uncontrolled logging on stream turbidity. Water and Sewage Works. July 1948.

Likens, G.E., F.H. Bormann, N.M. Johnson, D.W. Fisher, and R.S. Pierce. 1970. Effects of clearcutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem. Ecological Monographs, 40(1):23-47.

Lockaby, B. G., R. G. Clawson, K. Flynn, R. Rummer, S. Meadows, B. Stoke, and J. Stanturf. 1997. Influence of harvesting on biogeochemical exchange in sheetflow and soil processes in a eutrophic floodplain forest. Forest Ecology and Management. 90:187-194.

Lynch, J.A. and E.S. Corbett. 1990. Evaluation of best management practices for controlling nonpoint source pollution from silvicultural operations. Water Resources Bulletin. 26(1):41-52.

Lynch, J.A., E.S. Corbett, and K. Mussallem. 1985. Best management practices for controlling nonpoint source pollution on forested watersheds. Journal of Soil and Water Conservation. 40:164-167.

Martin, C.W., D.S. Noel, and C.A. Federer. 1984. Effects of forest clearcutting in New England on stream chemistry. Journal of Environmental Quality. 13:204-210.

McMinn, J.W. 1984. Soil disturbance by fuelwood harvesting with a conventional ground system and a cable miniyarder in mountain hardwood stands. pp. 93-98. In: Peters, P.A. and J. Luchok, eds., Proceedings from Mountain Logging Symposium. Charleston, WV: West Virginia University.

Megahan, W.F. 1980. Nonpoint source pollution from forestry activities in the Western United States: results of recent research and research needs. pp. 92-151. In: U.S. forestry and water quality: What course in the 80s? Proceedings of the Water Pollution Control Federation Seminar. Richmond, VA:

Miller, E.L., R.S. Beasley, and E.R. Lawson. 1988. Forest harvest and site preparation effects on erosion and sedimentation in the Ouachita Mountains. Journal of Environmental Quality. 17:219-225.

Miwa, M., W.M. Aust, J.A. Burger, Y.-J. Xu, and S.C. Patterson. 1999. Harvesting and site preparation effects on the hydroperiods of wet flat pine plantations. In: Proceedings from the Tenth Biennial Southern Silvicultural Research Conference, Shreveport, LA.

Mostaghimi, S., T.M. Wynn, J.W. Frazee, P.W. McClellan,, R.M. Shaffer, and W.M. Aust. 1999. Effects of forest harvesting best management practices on surface water quality in the Virginia Coastal Plain. Report No. FNC0999. Blacksburg, VA: Biological Systems Engineering, Virginia Tech.

Murphy, M.L. and J.D. Hall. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. Canadian Journal of Fisheries and Aquatic Science. 38:137-145.

Murphy, M.L., C.P. Hawkins, and N.H. Anderson. 1981. Effects of canopy modification and accumulated sediment on stream communities. Transactions of the American Fishery Society, 110:469-478.

National Council For Air And Stream Improvement. 1994. Forests as nonpoint sources of pollution, and effectiveness of best management practices. Tech. Bull. No. 672. New York, NY: National Council For Air And Stream Improvement.

National Council For Air And Stream Improvement.. 1999. Silviculture and water quality: a quarter century of Clean Water Act progress. Special Report No. 99-06. Research Triangle Park, NC: National Council For Air And Stream Improvement.

Neary, D.G., W.T. Swank, and H. Riekerk. 1989. An overview of nonpoint source pollution in the southern United States. In: Hook, Donal D. and Russ Lea, eds., Proceedings of the symposium: The forested wetlands of the Southern United States; 1988 July 12-14; Orlando, FL. Gen. Tech. Rep. SE-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 1-7.

Patric, J.H. 1976. Soil erosion in the eastern forest. Journal of Forestry. 74:671-677.

Patric, J.H. 1978. Harvesting effects on soils and water in the eastern hardwood forest. Southern Journal of Applied Forestry. 2(3): 66-73.

Plamondon, A.P., A. Gonzalez, and Y. Thomassin. 1982. Effects of logging on water quality: comparison between two Quebec sites. pp. 16-25. In: Canadian Hydrologic Symposium '82. National Research Council Canada, Associate Committee on Hydrology.

Reed, R.J. 1966. Some effects of DDT on the ecology of salmon streams in southeastern Alaska. Special Science Report, Fisheries 542. Washington DC: U.S. Fish and Wildlife Service.

Reid, L.M. 1993. Research and cumulative watershed effects. Gen. Tech. Rep. PSW-GTR-141. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture. 118pp.

Reid, L.M. and T. Dunne. 1984. Sediment production from forest road surfaces. Water Resources Research. 20:1753-1761.

Rice, R. M. and J.R. Wallis. 1962. How a logging operation can affect streamflow. Forest Industries. 89(11):38-40.

Richter, D. 2000. Soil and water effects of modern forest harvest practices in North Carolina. Individual White Paper associated with the "Economic and Ecologic Impacts Associated with Wood Chip Production in North Carolina Study," Durham, NC: Southern Center for Sustainable Forests, Duke University. http://www.env.duke.edu/scsf/. Accessed February 5, 2001.

Riekerk, H. 1983. Impacts of silviculture on flatwoods runoff water quality, and nutrient budgets. Water Resources Bulletin. 19:73-79.

Riekerk, H. 1985. Water quality effects of pine flatwoods silviculture. Journal of Soil and Water Conservation. 40(3):306-309.

Riekerk, H., D.G. Neary, and W.T. Swank. 1989. The magnitude of upland silvicultural nonpoint source pollution in the South. In: Hook, Donal D. and Russ Lea, eds., Proceedings of the symposium: The forested wetlands of the Southern United States; 1988 July 12-14; Orlando, FL. Gen. Tech. Rep. SE-50. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 1-7.

Rothwell, R.L. 1978. Watershed management guidelines for logging and road construction in Alberta. Information Report NOR-X-208. Alberta, Canada: Canadian Forestry Service, Northern Forest Research Centre.

Society of American Foresters. 2000. A review of waterbodies listed as impaired by silvicultural operations. SAF-00-03. Washington, DC: Society of American Foresters. In cooperation with the National Association of State Foresters. 13 p.

Scoles, S, S. Anderson, D. Turton, and E. Miller. 1996. Forestry and water quality: a review of watershed research in the Ouachita Mountains. Circular E-932, Water Quality Series. Stillwater, OK:Oklahoma Cooperative Extension Service, Division of Agricultural Sciences and Natural Resources, Oklahoma State University. 29 pp.

Segal, D.S., D.G. Neary, G.R. Best, and J.L. Michael. 1987. Effect of ditching, fertilization, and herbicide application on groundwater levels and groundwater quality in a flatwood spodosol. p.107-112. In: Proceedings of the Soil and Crop Science Society of Florida, Vol. 46.

Shepard, J.P. 1994. Effects of forest management on surface water quality in wetland forests. Wetlands. 13(1):18-26.

Sidle, R.C. 1989. Cumulative effects of forest practices on erosion and sedimentation. pp. 108-112. In: Forestry on the frontier, Proceedings of the 1989 Society of American Foresters, September 24-27, Spokane, WA. Bethesda, MD: Society of American Foresters

Silsbee, D. and G. Larson. 1982. Water quality of streams in the Great Smoky Mountains National Park. Hydrobiologia. 89:97-115.

Stickney, P.L., L.W. Swift, Jr., W.T. Swank, comps. 1994. Annotated bibliography of publications on watershed management and ecological studies at Coweeta Hydrologic Laboratory, 1934-1994. Gen. Tech. Rep. SE-86. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 115 p.

Stringer, J. and A. Thompson. 2000. Comparison of forestry best management practices, Part I: Streamside management zones. Forest Landowner. May/June 2000:22-27.

Sullivan, K and S.H. Duncan. 1981. Sediment yield from road surfaces in response to truck traffic and rainfall. Weyerhaeuser Technical Report 042-4404-81-04. Centralia, WA: Weyerhaeuser Forestry Research Center.

Sullivan, K., T.E. Lisle, C.A. Dolloff, G.E. Grant, and L.M. Reid. 1987. Stream channels: the link between forests and fishes. p. 39-97. In: Salo, E.O. and T.W. Cundy, eds., Streamside management: forestry and fishery interactions, College of Forest Resources Contribution #57. Seattle, WA: University of Washington.

Swank, W.T. 1988. Stream chemistry responses to disturbance. pp. 339-357. In: Swank, W.T. and D.A. Crossley, eds., Forest Hydrology and Ecology at Coweeta. New York, NY: Springer-Verlag.

Swank, W.T., L.F. DeBano, and D. Nelson. 1989. Effects of timber management practices on soil and water. p. 79-106. In: Burns, Russel L., tech. comp. The scientific basis for silvicultural and management decisions in the national forest system. Gen. Tech. Rep. WO-55. Washington, DC: U.S. Department of Agriculture, Forest Service.

Swank, W.T. and J.E. Douglass. 1974. Streamflow greatly reduced by converting deciduous hardwood stands to pine. Science. 185:857-859.

Swank, W.T. and N.H. Miner. 1968. Conversion of hardwood-covered watersheds to white pine reduces water yield. Water Resources Research. 4:947-954.

Swank, W.T., L.W. Swift, and J.E. Douglass. 1988. Streamflow changes associated with forest cutting, species conversions, and natural disturbances. In: Swank, W.T. and D.A. Crossley, Jr., eds. Forest hydrology and ecology at Coweeta. New York, NY: Springer Verlag.

Swank, W.T. and J.M. Vose. 1994. Long-term hydrologic and stream chemistry responses of Southern Appalachian catchments following conversion from mixed hardwoods to white pine. pp. 164-172. In: Landolt, Ruth, ed. Hydrologie kleiner Einzugsgebiete: Gedenkschrift Hans M. Keller. Beitrage zur Hydrologie der Schweiz 35. Bern, Schweizerische: Schweizerische Gesellschaft fur Hydrologie und Limnologie.

Swift, L.W., Jr. 1984a. Soil losses from roadbeds and cut and fill slopes in the Southern Appalachian Mountains. Southern Journal of Applied Forestry 8:209-216.

Swift, L.W., Jr. 1984b. Gravel and grass surfacing reduces soil loss from mountain roads. Forest Science. 30(3):657-670.

Swift, L.W., Jr. 1985. Forest Road Design to Minimize Erosion in the Southern Appalachians. pp. 141-151. In: Blackmon, B.G. ed. Forestry and water quality: A Mid-South Symposium. Little Rock, AR: University of Arkansas Cooperative Extension Service.

Swift, L.W., Jr. 1988. Forest access roads: design, maintenance, and soil loss. In: Swank, W.T. and D.A. Crossley, Jr., eds. Forest hydrology and ecology at Coweeta. New York, NY: Springer Verlag.

Swift, L.W. Jr. and S.E. Baker. 1973. Lower water temperatures within a streamside buffer strip. Res. Note SE-193. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 7 pp.

Swift, L.W. Jr. and R.G. Burns. 1999. The three Rs of roads: redesign, reconstruction, and restoration. Journal of Forestry. 97(8):40-44.

Swift, L.W. Jr. and J.B. Messer. 1971. Forest cuttings raise temperatures of small streams in the southern Appalachians. Journal of Soil and Water Conservation. 26:111-116.

Swift, L.W. Jr. and W.T. Swank. 1981. Long term responses of streamflow following clearcutting and regrowth. Hydrological Sciences Bulletin. 26:245-256.

Tebo, L.B., Jr. 1955. Effects of siltation resulting from improper logging on the bottom fauna of a small trout stream in the southern Appalachians. Progressive Fish-Culturist. 17:64-70.

Ursic, S.J. 1991. Forestry practices and the water resource of the Upper Coastal Plain. Resource Report 6: 93-96. Gainesville, FL: School of Forest Resources and Conservation, University of Florida.

Ursic, S.J. and J.E. Douglass. 1979. The effects of forestry practices on water resources. In: Proceedings of the W. Kelly Mosley environmental forum; 1978 May; Auburn, AL:33-49. Auburn, AL: Auburn University Press.

USDA Forest Service. 1984. Regional guide for the South. Atlanta.GA: U.S. Department of Agriculture, Forest Service, Southern Region.

U.S. Environmental Protection Agency. 1986. Quality criteria for water: 1986. Washington, DC: U.S. Environmental Protection Agency, Office of Water Regulations and Standards. 402 pp.

_____. 1990. National water quality inventory: 1988 report to Congress. EPA 440-4-90-003. Washington, DC: U.S. Environmental Protection Agency. 187 pp.

_____. 1992. National water quality inventory: 1990 report to Congress. EPA 503/9-92/006. Washington, DC: U.S. Environmental Protection Agency. 174 pp.

_____. 1994. National water quality inventory: 1992 Report to Congress. Washington, DC: US

Environmental Protection Agency.

U.S. Environmental Protection Agency. 1994. National water quality inventory: 1992 report to Congress. EPA 841-R-94-001. Washington, DC: U.S. Environmental Protection Agency. 328 pp.

_____. 1996. National water quality inventory: 1994 report to Congress. EPA 841-R-95-005. Washington, DC: U.S. Environmental Protection Agency. 497 pp.

_____. 1998. National water quality inventory: 1996 report to Congress. EPA 841-R-97-008. Washington, DC: U.S. Environmental Protection Agency. 521 pp.

_____. 2000a. Nutrient criteria: Technical guidance manual. EPA-822-B-00-002. Washington, DC: United States Environmental Protection Agency. 256 pp.

_____. 2000b. National water quality inventory: 1998 report to Congress. EPA 841-R-00-001. Washington, DC:U.S. Environmental Protection Agency. 413 pp.

Van Lear, D.H., J.E. Douglass, S.K. Cox, and M.K. Auspurger. 1985. Sediment and nutrient export in runoff from burned and harvested pine watersheds in the South Carolina Piedmont. Journal of Environmental Quality. 14(2):169-174.

Vowell, J.L. In press. Using stream bioassessment to monitor best management practice effectiveness. Forest Ecology and Management

Waters, T.F. 1995. Sediment in streams: sources, biological effects, and control. American Fisheries Society Monograph 7. Bethesda, MD: American Fisheries Society. 251 pp.

Williams, T.M. and G.R. Askew. 1988. Impact of drainage and site conversion of pocosin lands on water quality. p. 213-218. In: D.D. Hook, ed., The ecology and management of wetlands, Vol. 2: Management, use, and value of wetlands. London, UK: Croom Held

Wu, Y-J., W.M Aust, J.A. Burger, S.C. Patterson, and M. Miwa. 1999. Recovery of hydroperiod after timber harvesting in a forested wetland. In: Proceedings of the Tenth Biennial Southern Silvicultural Research Conference; 1999 February 16-18; Shreveport, LA. Gen. Tech. Rep. SRS-30. Asheville, NC: U.S. Department of Agriculture, Forest Service: 282-287.

Wu, Y-J., W.M Aust, J.A. Burger, S.C. Patterson, and M. Miwa. 1999. Recovery of hydroperiod after timber harvesting in a forested wetland. In: Proceedings of the Tenth Biennial Southern Silvicultural Research Conference, Shreveport, LA.

Yoho, N.S. 1980. Forest management and sediment production in the South--a review. Southern Journal of Applied Forestry. 4:27-36.

9 Tables and Figures

Table 1--Total river miles impaired by silviculture in the South (1988-1998)

State	1988	1990	1992	1994	1996	1998	Average 1988-1998
	Impaired river miles[a]						
Alabama	0	196	218	195	219	0	138
Arkansas	0	261	193	251	218	0	154
Florida	63	142	154	1,181	1,410	428	563
Georgia	0	--	--	--	3	0	1
Kentucky	--	--	34	120	103	56	78
Louisiana	--	1,339	1,167	758	1,328	326	984
Mississippi	0	405	2,051	408	2,310	2,121	1,216
North Carolina	48	--	313	276	243	151	206
Oklahoma	20	--	126	126	110	218	120
South Carolina	4	--	--	326	221	221	193
Tennessee	140	142	--	74	524	61	188
Texas	--	--	--	--	--	0	0
Virginia	0	--	--	166	658	11	209
Total	275	2,485	4,256	3,881	7,347	3,593	3,639

[a] A river mile includes all river and stream miles reported by each State. An impaired river mile is classified as partially supporting overall use or not supporting overall use.

-- Not reported

Source: U.S. Environmental Protection Agency 1990, 1992, 1994, 1996, 1998, 2000b (National Water Quality Inventory Reports to Congress)

Return to first reference in text

Return to second reference in text

Return to third reference in text